

Table 4-5. Predator-prey relationships for species commonly impinged or entrained at the Pilgrim facility (cont.).

Species	Prey	Predators	References
Rock gunnel	Small crustaceans, polychaetes, molluscs and fish eggs.	Cod, pollock	Bigelow and Schroeder, 1953; Froese and Pauly, 2000
Shorthorn sculpin	Crab and other crustaceans, shrimp, sea urchins, worms, and fry of other fish.	Atlantic cod	Froese and Pauly, 2000; Bigelow and Schroeder, 1953
Silver hake	Fish (alewife, butterfish, cunner, herring, mackerel, menhaden, scup, silversides, smelt, young of its own species), crustaceans, shrimp.	Bluefish, butterfish	Bigelow and Schroeder, 1953; Morse et al., 1999
Striped bass	Mysid shrimp and smaller fish species such as herring, silversides, and anchovies; Larvae feed primarily on copepods.	Sea lamprey, striped bass, silver hake, bluefish, copepods	Miller, 1995
Striped killifish	Crustaceans and polychaetes.	Wading birds, aerial searching birds, piscivorous ducks, crabs, and many predatory fishes. Fishes include white perch, summer flounder, striped bass, bluefish, and red drum. Birds include herons, egrets, terns, gulls, and least common terns	Abraham, 1985
Summer flounder	Small fish, small shelled mollusks, worms, sand dollars, squids, crabs, shrimp, and other crustaceans.	Larvae and juveniles: spiny dogfish, cod, goosefish, hake, sea raven, longhorn sculpin, and fourspot flounder	Bigelow and Schroeder, 1953
Tautog	Mussels, small crustaceans and other molluscs. Juveniles feed on amphipods and copepods.	Smooth dogfish, barndoor skate, red hake, sea raven, goosefish, and seabirds	Jury et al., 1994; Steimle and Shaheen, 1999
Threespine stickleback	Omnivorous. Small invertebrates, fish fry, fish eggs, shrimp, small squids, and diatoms.	Sea trout, whiting, eels	Bigelow and Schroeder, 1953; Froese and Pauly, 2000
White perch	Variety of prey, including shrimp, fish, and crab. Their diet composition changes with seasonal and spatial food availability. Larvae feed mainly on plankton.	Striped bass, bluefish, weakfish, walleye, copepods	Beck, 1995

Table 4-5. Predator-prey relationships for species commonly impinged or entrained at the Pilgrim facility (cont.).

Species	Prey	Predators	References
Windowpane	Young consume mysids, while adults feed on sand shrimp, small fish (up to 10 cm), crustaceans, molluscs, and seaweed.	Spiny dogfish, thorny skate, goosefish, Atlantic cod, black sea bass, weakfish, and summer flounder	Chang et al., 1999
Winter flounder	Benthic organisms such as shrimp, amphipods, crabs, urchins and snails.	Larger estuarine and coastal fish such as striped bass and bluefish	Buckley, 1989b; Froese & Pauly, 2000
Yellowtail flounder	Small crustaceans (including amphipods, shrimps, and mysids), small shellfish, and worms.	Spiny dogfish, skates, Atlantic halibut, fourspot flounder, goosefish, silver hake, bluefish and sea raven	Bigelow and Schroeder, 1953; Johnson et al., 1999b

4.3 Step 3: Identify Potential Habitat Restoration Alternatives to Offset I&E Losses

Local experts proposed six types of habitat restoration projects that would offset I&E losses at the Pilgrim facility:

- ▶ improve water quality
- ▶ reduce fishing pressures
- ▶ restore tidal wetlands
- ▶ restore submerged aquatic vegetation
- ▶ improve anadromous fish passage
- ▶ create artificial reefs.

Each of these potential restoration projects provide benefits to the aquatic community, and are described below.

Improve water quality

Water quality plays a major role in determining whether fish can survive in a given water body. Water quality can be compromised by high levels of industrial pollutants, nutrients from wastewater treatment plants and failing septic systems, and extreme temperatures. Some examples of water quality improvement projects may include (but are not limited to):

- ▶ remove nitrogen and phosphorus at wastewater treatment plants
- ▶ improve storm water management

- ▶ repair or replace failing septic systems
- ▶ provide better “pump-out” services to recreational and commercial boaters to dispose of their boat waste in a safe and sanitary manner
- ▶ limit discharges of hazardous materials from industrial facilities
- ▶ limit thermal discharges.

Any measures to improve water quality by limiting the amount of pollutants in the estuaries surrounding the Pilgrim facility benefit the aquatic ecosystem. Reducing pollutant levels will increase survival rates for invertebrates, fish, and other animals that depend on the estuarine ecosystem. Improving water quality can restore fish and shellfish habitats that were previously limited or uninhabitable because of toxicity or intolerance to polluted conditions.

Reduce fishing pressures

Fish that support commercial or recreational fisheries are prone to high mortality rates because of fishing pressures. These species can benefit from reduced fishing. Some potential projects that could be implemented to reduce fishing pressures include closing sensitive areas (such as spawning grounds) to fishing during certain times during the year, or decreasing the number of fishing licenses that are issued. Fishing gear could also be changed to limit the number of unwanted fish caught. For example, fishing nets could be altered to reduce the catch of small or undesirable fish that are caught in existing nets.

Restore tidal wetlands

Tidal wetlands (Figure 4-1) are among the most productive ecosystems in the world (Mitsch and Gosselink, 1993; Broome and Craft, 2000). Tidal wetlands provide valuable habitat for many species of invertebrates and forage fish that serve as food for other species in and near the wetland. Tidal wetlands also provide spawning and nursery habitat for many other fish species, including the Atlantic silverside, striped killifish, threespine stickleback, and mummichog. Other migratory species that use tidal wetlands during their lives include the winter flounder, striped bass, Atlantic herring, and white perch (Dionne et al., 1999). Fish species that have been reported in restored salt ponds and tidal creeks include Atlantic menhaden, blueback herring, Atlantic silverside, striped killifish, and mummichog [Roman et al., (submitted to *Restoration Ecology*)]. Restoring tidal flow to areas where such flows have been restricted has also been shown to reduce the presence of *Phragmites australis*, the invasive marsh grass that has choked out native flora and fauna in coastal areas across the New England seaboard (Fell et al., 2000).

Tidal wetlands restoration typically involves returning tidal flow to marshes or ponds that have restrictions of natural tidewater flow by roads, backfilling, dikes, or other barriers. Eliminating these barriers can restore salt marshes (Figure 4-2), salt ponds, and tidal creeks that provide essential habitat for many species of aquatic organisms. For example, where tidal flow is reduced



Figure 4-1. Tidal creek near Little Harbor, Cohasset, Massachusetts.

Source: MAPC, 2001.



Figure 4-2. Salt marsh near Narragansett Bay, Rhode Island.

Source: Save The Bay, 2001.

by undersized culverts, installing correctly sized and positioned culverts can restore tidal range and proper salinity. In other situations, such as where low-lying property adjacent to salt marsh has been developed, restoring full tidal flow may not be possible because of flood concerns (MAPC, 2001). Salt marshes can also be created by flooding areas in which no marsh habitat previously existed (e.g., tidal wetland creation). However, a study by Dionne et al. (1999) showed that while both created and restored tidal wetlands readily provide habitat for a number of fish, restored tidal wetlands provide much larger and more productive areas of habitat per unit cost than created tidal wetlands.

Restore submerged aquatic vegetation

Submerged aquatic vegetation (SAV) provides vital habitat for a number of aquatic organisms. Eelgrass is the dominant species of SAV along the coasts of New England. It is an underwater flowering plant that is found in brackish and near-shore marine waters (Figure 4-3). Eelgrass can form large meadows or small separate beds that range in size from many acres to just 1 m across (Save The Bay, 2001).



Figure 4-3. Laboratory culture of eelgrass (*Zostera marina*).

Source: Boschker, 2001.

SAV restoration involves transplanting eelgrass shoots and/or seeds into areas that can support their growth. Site selection is based on historical distribution, wave action, light availability, sediment type, and nutrient loading. Improving water quality and clarity, reducing nutrient levels, and restricting dredging may all be necessary to promote sustainable eelgrass beds. Protecting existing SAV beds is a priority in many communities (Save The Bay, 2001).

SAV provides several ecological services to the environment. It has a high rate of leaf growth and provides support for many aquatic organisms as shelter, spawning, and nursery habitat. It is also a food source for herbivorous organisms. The roots of SAV also provide stability to the bottom sediments, thus decreasing erosion and resuspension of sediments into the water column (Thayer et al., 1997). Dense SAV provides shelter for small and juvenile fishes and invertebrates from predators. Small prey can hide deep within the SAV canopy, and some prey species use the SAV as camouflage (Thayer et al., 1997). Species that use SAV beds during early life stages include Atlantic menhaden, striped bass, American eel, tautog, bluefish, summer flounder, weakfish, rainbow smelt, bay scallops, and blue crab (Laney, 1997).

Improve anadromous fish passageways

Anadromous fish spend most of their lives in brackish or saltwater but migrate into freshwater rivers and streams to spawn. Many of the rivers and streams that historically supported anadromous fish spawning have been dammed and are currently inaccessible to migrating fish. Anadromous fish that would benefit from improved access to upstream spawning habitat include alewife, Atlantic salmon (*Salmo salar*), rainbow smelt, sturgeon, white perch, American eel, and American shad.

Improving anadromous fish passage involves many important steps. Dams and barriers connecting estuaries with upstream spawning habitat can be removed or fitted with fish ladders (Figure 4-4). Removing the dam is often preferable because some species, such as rainbow smelt, use fish ladders ineffectively. However, dam removal may not be possible in highly developed areas needing flood control. In addition, restoring stream habitats such as forested riverbank wetlands and improving water quality may also be necessary to restore upstream spawning habitats for anadromous fish (Save The Bay, 2001).

Create artificial reefs

Several species of fish found near the Pilgrim facility use rocky or reef-like habitats with interstices that provide refuge from predators. These habitats can be created artificially with cobbles, concrete, and other suitable materials.

Species that commonly use reef structures for refuge include tautog, cunner, scup, black sea bass, lobsters, and blue mussels (Foster et al., 1994; Castro et al., in press). Both cunner and tautog become torpid at night and require places to hide from their prey. Blue mussels use rocky reefs for attachment.

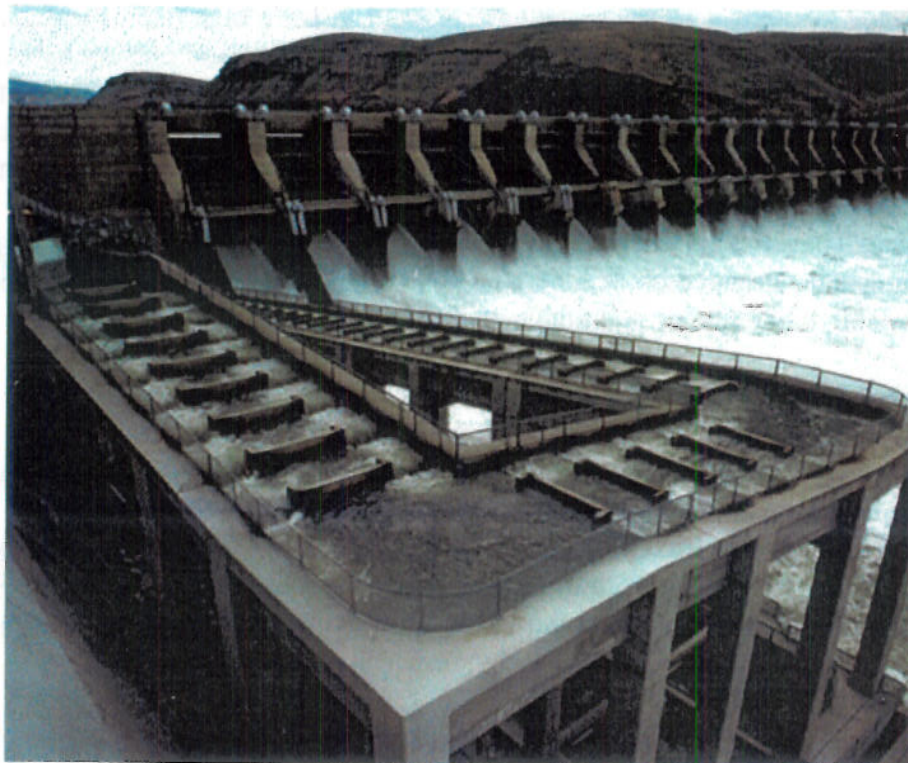


Figure 4-4. Example of a fish ladder at a hydroelectric dam.

Source: Pollock, 2001.

4.4 Step 4: Consolidate, Categorize, and Prioritize Identified Habitat Restoration Alternatives

Habitat restoration alternatives were categorized and prioritized in collaboration with local experts. Meetings were designed to identify the restoration program for each of the major species that are impinged or entrained as a result of cooling water intakes. Meetings were arranged and moderated by Stratus Consulting, and attended by several federal, state, and local organizations (Table 4-6).

Habitat needs and restoration options for each species with significant I&E losses at the facility were discussed. These restoration options were then prioritized for each species by determining what single restoration option would most benefit that species. The higher ranked restoration alternatives for each species are shown in Table 4-7.

Table 4-6. Attendees at the Pilgrim Meeting, September 12, 2001, in Lakeville, Massachusetts.

Attendee	Organization
David Allen	Stratus Consulting
David Mills	Stratus Consulting
Michelle Barron	Stratus Consulting
Bob Green	Massachusetts DEP
Robert Lawton	Massachusetts Division of Marine Fisheries
George Zoto	Massachusetts Watershed Initiative - South Coastal Watersheds
Kathi Rodrigues	National Marine Fisheries Service - Restoration Center
David Webster	U.S. EPA Region I
Sharon Zaya	U.S. EPA Region I
Nick Prodan	U.S. EPA Region I
John Nagle	U.S. EPA Region I

Table 4-7. Restoration alternatives for each Pilgrim species ranked highest by local experts.

Species	Prioritized restoration alternatives
Alewife	Anadromous fish passage
Atlantic herring	Anadromous fish passage
Blueback herring	Anadromous fish passage
Rainbow smelt	Anadromous fish passage (remove dams)
White perch	Anadromous fish passage
Cunner	Artificial reefs, SAV restoration
Sculpin spp.	Artificial reefs, SAV restoration (improve habitat for prey)
Tautog	Artificial reefs, SAV restoration
American sand lance	Tidal wetlands restoration
Atlantic silverside	Tidal wetlands restoration
Bluefish	Tidal wetlands restoration (improve habitat for prey)
Grubby	Tidal wetlands restoration
Striped bass	Tidal wetlands restoration (improve habitat for prey)
Windowpane ^a	Tidal wetlands restoration (improve habitat for prey)
Winter flounder	Tidal wetlands restoration
Threespine stickleback	SAV restoration, tidal wetland restoration
Atlantic mackerel	Reduce fishing pressure, improve water quality
Atlantic menhaden	Reduce fishing pressure, improve water quality
Bay anchovy	Reduce fishing pressure, improve water quality
Butterfish	Reduce fishing pressure, improve water quality
All species	Improve water quality

a. Improved water quality later became the chosen restoration alternative for windowpane because they inhabit depths greater than accessible to tidal wetland restoration.

Table 4-12. Average abundance from Rhode Island SAV sites for Pilgrim species that would benefit most from SAV restoration.

Species	Species abundance (# fish per 100 m ² of SAV habitat) ^a	
	Low quality SAV habitats	High quality SAV habitats
Atlantic tomcod	0.52	1.77
Pollock	no obs.	no obs.
Northern pipefish	0.23	3.03
Threespine stickleback	no obs.	19.67

a. High quality habitats are defined as areas with eelgrass shoot densities > 100 per m² and shoot biomass (wet) > 100 g/m². Low quality habitats do not meet these criteria.

Source: personal communication, J. Hughes, NOAA, Marine Biological Laboratory, 2001.

Heck et al., 1989 — Species abundance in Nauset Marsh (Massachusetts) estuarine complex SAV

Heck et al. (1989) provide capture totals for day and night trawl samples taken between August 1985 and October 1986 in the Nauset Marsh Estuarine Complex in Orleans/Eastham, Massachusetts, including two eelgrass beds: Fort Hill and Nauset Harbor. As in the other SAV sampling efforts, an otter trawl was used for the sampling, but with slightly larger mesh size openings in the cod end liner (6.3 mm versus 3.0 mm) than in Hughes et al. (2001) or Wyda et al. (in press).

With the reported information on the average speed, duration, and number of trawls used in each sampling period and an estimate of the width of the SAV habitat covered by the trawl from one of the study authors (personal communication, M. Fahay, NOAA, 2001), abundance estimates per 100 m² of SAV habitat were calculated.

Heck et al. (1989) also report that the dry weight of the SAV shoots is over 180 g/m² at both the Fort Hill and Nauset Harbor eelgrass habitat sites. Therefore, these locations would fall into the high SAV habitat category used in Wyda et al. (in press) and Hughes et al. (2000) because the dry weight exceeds the wet weight criterion of 100 g/m² used in those studies.

Finally, Heck et al. (1989) provide separate monthly capture results from their trawls. The maximum monthly capture results for each species was used for the abundance estimates from this sampling. Because these maximum values generally occur in the late summer months, sampling time is consistent with the results from Wyda et al. (in press) and Hughes et al. (2000).

The species abundance values estimated from the sampling of the Fort Hill and Nauset Harbor SAV habitats are presented in Table 4-13.

Table 4-13. Average abundance in Nauset Marsh Estuarine Complex SAV for Pilgrim species that would benefit most from SAV restoration.

Species	Species abundance (# fish per 100 m ²) ^a	
	Fort Hill — High quality SAV	Nauset Harbor — High quality SAV
Atlantic tomcod	no obs.	0.08
Pollock	no obs.	no obs.
Northern pipefish	0.68	6.11
Threespine stickleback	5.92	47.08

a. High quality habitats are defined as areas with eelgrass shoot densities > 100 per m² and shoot biomass (wet) > 100 g/m².

Source: Heck et al., 1989.

4.5.1.2 Adjusting SAV sampling results to estimate annual average increase in production of age-1 fish

Sampling-based abundance estimates were adjusted to account for:

- ▶ sampling efficiency
- ▶ capture of life stages other than age 1
- ▶ differences in the productivity of restored versus natural SAV habitat.

The basis and magnitude of the adjustments are discussed in the following sections.

Adjusting for sampling efficiency

Fish sampling techniques are unlikely to capture and/or record all of the fish present in a sampled area because some fish avoid the sampling gear and some are captured but not collected and counted. The sampling efficiency for otter trawls is approximately 40% to 60% (personal communication, J. Hughes, NOAA Marine Biological Laboratory, 2001). A conservative sampling efficiency of 40% was assumed for this HRC analysis. Therefore, the SAV sampling abundance estimates were multiplied by 2.5 (i.e., divided by 40%). This assumption increases SAV productivity estimates and lowers SAV restoration cost estimates.

Adjusting sample abundance estimates to age-1 life stages

All sampled life stages were converted to age-1 equivalents for comparison to I&E losses, which were expressed as age-1 equivalents. The average life stage of the fish caught in the Buzzards Bay (Wyda et al., in press) and Rhode Island coastal salt pond (Hughes et al., 2000) was juveniles (i.e., life stage younger than age 1) (personal communication, J. Hughes, NOAA Marine Biological Laboratory, 2001). Since the same sampling technique and gear was used in Heck et al. (1989), juveniles were assumed to be the average life stage captured in this study as well.

Table 4-15. Final estimates of the increase in production of age-1 fish for Pilgrim species that would benefit most from SAV restoration (cont.).

Species	Source of initial species abundance estimate	Species abundance estimate per 100 m ² of SAV	Sampling efficiency adjustment factor	Life stage adjustment factor	Restored habitat service flow adjustment factor	Expected increase in production of age-1 fish per 100 m ² of restored SAV
Northern pipefish	Hughes et al. (2000) — RI coastal ponds (high SAV)	3.03	2.5	0.5352	1.0	4.06
	Wyda et al. (in press) — Buzzards Bay (low SAV)	0.19	2.5	0.5352	1.0	0.25
	Wyda et al. (in press) — Buzzards Bay (high SAV)	0.99	2.5	0.5352	1.0	1.32
	Species average					2.50
Threespine stickleback	Heck et al. (1989) — Fort Hill	5.92	2.5	0.5284	1.0	7.82
	Heck et al. (1989) — Nauset Harbor	47.08	2.5	0.5284	1.0	62.19
	Hughes et al. (2000) — RI coastal ponds (high SAV)	19.67	2.5	0.5284	1.0	25.98
	Wyda et al. (in press) — Buzzards Bay (low SAV)	0.22	2.5	0.5284	1.0	0.29
	Wyda et al. (in press) — Buzzards Bay (high SAV)	0.13	2.5	0.5284	1.0	0.17
	Species average					19.29
Pollock	no obs.					

4.5.2 Estimates of Increased Age-1 Fish Production from Tidal Wetland Restoration

Tidal wetlands provide a diversity of habitats such as open water, subtidal pools, ponds, intertidal waterways, and tidally flooded meadows of salt tolerant species such as *Spartina alterniflora* and *S. patens*. These habitats provide forage, spawning, nursery, and refuge for a large number of fish species. Table 4-16 identifies the I&E losses for fish species at Pilgrim that would benefit most from tidal wetland restoration, along with average I&E losses for the period 1974-1999, arranged by number of fish lost.

Table 4-16. Pilgrim species that would benefit most from tidal wetland restoration.

Species	Annual average I&E loss of age 1 equivalents (1974-1999)	Percentage of annual average I&E loss across all fish species
American sand lance	4,116,285	28.55%
Winter flounder	210,715	1.46%
Atlantic silverside	25,929	0.18%
Grubby	879	0.01%
Striped killifish	90	0.00%
Striped bass	9	0.00%
Bluefish	2	0.00%
Total	4,353,909	30.20%

Restricted tidal flows increase the dominance of *Phragmites australis* by reducing tidal flushing and lowering salinity levels (Buzzards Bay Project National Estuary Program, 2001). *Phragmites* dominance restricts fish access to and movement through the water, decreasing overall productivity of the habitat. Therefore, for the purpose of this HRC valuation, tidal wetland restoration focuses on returning natural tidal flows to currently restricted areas. Examples of actions that can restore tidal flows to currently restricted tidal wetlands include the following:

- ▶ breaching dikes created to support salt hay farming or to control mosquitos
- ▶ installing properly sized culverts in areas currently lacking tidal exchange
- ▶ removing tide gates on existing culverts
- ▶ excavating dredge spoil covering former tidal wetlands.

No identified studies quantified increased production following implementation of these types of restoration actions for tidal wetlands. Therefore, fish abundance estimates taken from studies of tidal wetlands were used to estimate the fish increase in production that can be gained through restoration. The following subsections present the sampling data and subsequent adjustments made to calculate the expected increased in age-1 production of fish species.

4.5.2.1 Fish species abundance estimates in tidal wetlands habitats

Results from tidal wetland sampling efforts in Rhode Island were used to calculate increased production. Available sampling results from Connecticut (Warren et al., submitted to *Restoration Ecology*) and New Hampshire and Maine coasts (Dionne et al., 1999) were not used. The Connecticut results were omitted because time constraints prevented the conversion of capture results into abundance estimates per unit of tidal wetland area. The New Hampshire and Maine results were omitted because the study locations were too distant from the Pilgrim facility.

Roman et al. (submitted to *Restoration Ecology*) — Species abundance at Sachuest Point tidal wetland, Middletown, Rhode Island

Roman et al. (submitted to *Restoration Ecology*) sampled the fish populations in a 6.3 ha unrestricted tidal wetland at Sachuest Point in Middletown, Rhode Island. The sampling was conducted during August, September, and October of 1997, 1998, and 1999 using a 1 m² throw trap in the creeks and pools of each area during low tide after the wetland surface had drained. Additional sampling was conducted monthly in both the unrestricted and restricted parcels from June through October in 1998 and 1999 using 6 m² bottomless lift nets to sample the flooded wetland surface. The report presents the results of this sampling as abundance estimates of each fish species per square meter (Table 4-17).

Table 4-17. Abundance estimates from the unrestricted tidal wetlands at Sachuest for Pilgrim species that would benefit most from tidal wetlands restoration.

Species	Sampling technique	Fish density estimates in unrestricted tidal wetlands (fish per m ²)		
		1997	1998	1999
American sand lance	throw trap	no obs.	no obs.	no obs.
	lift net	no sampling	no obs.	no obs.
Winter flounder	throw trap	no obs.	no obs.	no obs.
	lift net	no sampling	no obs.	no obs.
Atlantic silverside	throw trap	1.23	0.20	0.07
	lift net	no sampling	no obs.	no obs.
Grubby	throw trap	no obs.	no obs.	no obs.
	lift net	no sampling	no obs.	no obs.
Striped killifish	throw trap	0.70	0.17	0.55
	lift net	no sampling	0.01	0.01
Striped bass	throw trap	no obs.	no obs.	no obs.
	lift net	no sampling	no obs.	no obs.
Bluefish	throw trap	no obs.	no obs.	no obs.
	lift net	no sampling	no obs.	no obs.

Source: Roman et al. (submitted to *Restoration Ecology*).

Roman et al. (submitted to *Restoration Ecology*) also sampled a smaller portion of the wetland where tidal flows had recently been restored. However, these results were not used because the sampling most likely was conducted prior to the system reaching full productivity.

Raposa (in press) — Galilee Marsh, Naragansett Rhode, Island

Raposa (in press) sampled the fish populations in the Galilee tidal wetland monthly from June through September of 1997, 1998, and 1999 using 1 m² throw trap in the creeks and pools in the tidal wetland parcels during low tide after the wetland surface had drained. Raposa presents the sampling results as fish species abundance expressed as number of fish per square meter. As with the results from Roman et al. (submitted to *Restoration Ecology*), results from a recently restored portion of the wetland were not used in this HRC to avoid a downward bias in the species density results. The results from this sampling effort are presented in Table 4-18 for the Pilgrim species that would benefit most from tidal wetlands restoration.

Table 4-18. Abundance estimates from the unrestricted tidal wetlands at Galilee for Pilgrim species that would benefit most from tidal wetland restoration.

Species	Sampling technique	Fish density estimates in unrestricted tidal wetlands (fish per m ²)		
		1997	1998	1999
American sand lance	throw trap	no obs.	no obs.	no obs.
Winter flounder	throw trap	no obs.	no obs.	no obs.
Atlantic silverside	throw trap	4.78	1.73	14.38
Grubby	throw trap	no obs.	no obs.	no obs.
Striped killifish	throw trap	4.35	3.50	12.40
Striped bass	throw trap	no obs.	no obs.	no obs.
Bluefish	throw trap	no obs.	no obs.	no obs.

Source: Raposa, in press.

K. Raposa, Naragansett Estuarine Research Reserve, personal communication, 2001 — Coggeshall Marsh, Prudence Island, Rhode Island

Discussions with Kenny Raposa of the Naragansett Estuarine Research Reserve (NERR) revealed that additional fish abundance estimates from tidal wetland sampling were available for the Coggeshall Marsh located on Prudence Island in the NERR. These abundance estimates were based on sampling conducted in July and September 2000. The sampling of the Coggeshall tidal wetland was conducted using 1 m² throw traps in the tidal creeks and pools of the wetland during ebb tide after the wetland surface had drained (personal communication, K. Raposa, Naragansett Estuarine Research Reserve, 2001). The sampling results from this effort are presented in Table 4-19 for the Pilgrim species that would benefit most from tidal wetlands restoration.

The sampling efficiencies of bottomless lift nets for individual fish species are provided in Rozas (1992), and are 93% for striped mullet (*Mugil cephalus*), 81% for gulf killifish (*Fundulus grandis*), and 58% for sheepshead minnow (*Cyprinodon variegatus*). The average of these three sampling efficiencies is 77%, which corresponds to a sampling efficiency adjustment factor of 1.3 (i.e., $1.0/0.77$).

Lastly, although specific studies of the sample efficiency of a beach seine net were not identified, an estimated range of 50% to 75% was provided by the staff involved with the Rhode Island coastal pond survey (personal communication, J. Temple, Rhode Island Division of Fish and Wildlife, 2002). Using the lower end of this range as a conservative assumption, a sample efficiency adjustment factor of 2.0 (i.e., $1.0/0.5$) was applied for the abundance estimates for both the Rhode Island juvenile finfish survey and the Rhode Island coastal pond survey.

Conversion to age-1 life stage

The sampling techniques described in Section 4.5.2.1 are intended to capture juvenile fish (personal communication, K. Raposa, Naragansett Estuarine Research Reserve, 2001). That juvenile fish were the dominant age class taken was confirmed by the researchers involved in these efforts (personal communication, K. Raposa, Naragansett Estuarine Research Reserve, 2001; personal communication, C. Powell, Rhode Island Department of Environmental Management, 2001; and personal communication, J. Temple, Rhode Island Division of Fish and Wildlife, 2001). As a result, the sampling results presented in Section 4.5.2.1 required adjustment to account for expected mortality between the juvenile and age-1 life stages. The information used to develop these survival rates and the final life stage adjustment factors are presented in Table 4-22.

Table 4-22. Life stage adjustment factors for Pilgrim species — Tidal wetland restoration.

Species	Oldest life stage before age 1 in I&E model	Estimated survival rate to age 1	Life stage captured in tidal wetland sampling efforts	Estimated life stage adjustment factor
American sand lance	larvae	0.0298	juvenile	0.5149
Winter flounder	juvenile	0.2903	juvenile	0.2903
Atlantic silverside	larvae	0.0044	juvenile	0.5022
Grubby	larvae	0.0180	juvenile	0.5090
Striped killifish	larvae	0.0949	juvenile	0.5474
Striped bass ^a	juvenile	0.5361	juvenile	0.5361
Bluefish	juvenile	0.0103	juvenile	0.0103

a. Information in the I&E model is available for two juvenile life stages for striped bass. The data for the older juvenile life stage were used.

Adjusting for differences between restored and undisturbed habitats

Restoring full tidal flows rapidly eliminates differences in fish populations between unrestricted and restored sites (Roman et al., submitted to *Restoration Ecology*), resulting in very similar species composition and density (Dionne et al., 1999; Fell et al., 2000; Warren et al., submitted to *Restoration Ecology*). However, a lag can occur following restoration (Raposa, in press). Therefore, an adjustment factor of 1.0 was used, signifying that no quantitative adjustment was necessary.

Adjusting sampled abundance for timing and location of sampling

At high tide, fish in a tidal wetland have access to the full range of habitats, including the flooded vegetation, ponds, and creeks that discharge into or drain the wetland. In contrast, at low tide, fish are restricted to tidal pools and creeks. Therefore, sampling conducted at low tide represents a larger area of tidal wetlands than the sampled area. Abundance estimates based on samples taken at low tide were therefore divided by the inverse of the proportion of subtidal habitat to total wetland habitat. In contrast, no adjustment was applied to abundance estimates based on samples such as those from lift nets or seines, taken at high tide or in open water offshore. The site-specific adjustment factors in Table 4-23 were based on information regarding the proportion of each tidal wetland that is subtidal habitat (personal communication, K. Raposa, Naragansett Estuarine Research Reserve, 2001).

Table 4-23. Adjustment factors for tidal wetland sampling conducted at low tide.

Tidal wetland	Ratio of open water (creeks, pools) to total habitat in the wetland	Adjustment factor
Sachuest Marsh	0.055	18.2
Galilee Marsh	0.084	11.9
Coggeshall Marsh	0.052	19.2

4.5.2.3 Final estimates of annual average age-1 fish production from tidal wetland restoration

Table 4-24 presents the final estimates of annual increased production of age-1 fish resulting from tidal wetland restoration for Pilgrim species that would benefit most from tidal wetland restoration.

Table 4-24. Final estimates of the annual increase in production of age-1 equivalent fish per square meter of restored tidal wetland for Pilgrim species that would benefit most from tidal wetland restoration.

Species	Source of initial species density estimate	Sampling location and date ^a	Reported/ calculated species density estimate per m ² of tidal wetland	Sampling efficiency adjustment factor	Life stage adjustment factor	Restored habitat service flow adjustment factor	Sampling time and location adjustment factor	Increased production of age 1 fish per m ² of restored tidal wetland ^b
American sand lance	no obs.							
Winter flounder	Raposa pers comm 2001	NERR — Prudence Isl. Coggeshall - July 2000	0.10	1.6	0.2903	1	19.23	0.00
	Raposa pers comm 2001	NERR — Prudence Isl. Coggeshall — Sept. 2000	0.10	1.6	0.2903	1	19.23	0.00
	C Powell pers comm 2001	Chepiwanoxet average 1990-2000 (seine)	0.09	2.0	0.2903	1	1.00	0.05
	C Powell pers comm 2001	Wickford average 1990-2000 (seine)	0.20	2.0	0.2903	1	1.00	0.12
	J. Temple pers comm 2002	Narrow River average 1998-2001 (seine)	0.32	2.0	0.2903	1	1.00	0.19
	J. Temple pers comm 2002	Winnapaug Pond average 1998-2001 (seine)	0.21	2.0	0.2903	1	1.00	0.12
	J. Temple pers comm 2002	Point Judith Pond average 1998-2001 (seine)	0.21	2.0	0.2903	1	1.00	0.12
	Species average							0.09

Table 4-24. Final estimates of the annual increase in production of age-1 equivalent fish per square meter of restored tidal wetland for Pilgrim species that would benefit most from tidal wetland restoration (cont.).

Species	Source of initial species density estimate	Sampling location and date ^a	Reported/calculated species density estimate per m ² of tidal wetland	Sampling efficiency adjustment factor	Life stage adjustment factor	Restored habitat service flow adjustment factor	Sampling time and location adjustment factor	Increased production of age 1 fish per m ² of restored tidal wetland ^b
Atlantic silverside	Roman et al., submitted to <i>Restoration Ecology</i>	Sachuest Point — 1997	1.23	1.6	0.5022	1	18.18	0.05
	Roman et al., submitted to <i>Restoration Ecology</i>	Sachuest Point — 1998	0.20	1.6	0.5022	1	18.18	0.01
	Roman et al., submitted to <i>Restoration Ecology</i>	Sachuest Point — 1999	0.07	1.6	0.5022	1	18.18	0.00
	Raposa pers comm 2001	NERR — Prudence Isl. Coggeshall - July 2000	0.17	1.6	0.5022	1	19.23	0.01
	Raposa pers comm 2001	NERR — Prudence Isl. Coggeshall — Sept. 2000	0.07	1.6	0.5022	1	19.23	0.00
	Raposa, in press	Galilee Marsh — 1997	4.78	1.6	0.5022	1	11.90	0.32
	Raposa, in press	Galilee Marsh — 1998	1.73	1.6	0.5022	1	11.90	0.12
	Raposa, in press	Galilee Marsh — 1999	14.38	1.6	0.5022	1	11.90	0.97
	Species average							0.19

Table 4-24. Final estimates of the annual increase in production of age-1 equivalent fish per square meter of restored tidal wetland for Pilgrim species that would benefit most from tidal wetland restoration (cont.).

Species	Source of initial species density estimate	Sampling location and date ^a	Reported/ calculated species density estimate per m ² of tidal wetland	Sampling efficiency adjustment factor	Life stage adjustment factor	Restored habitat service flow adjustment factor	Sampling time and location adjustment factor	Increased production of age 1 fish per m ² of restored tidal wetland ^b
Grubby	no obs.							
Striped killifish	Roman et al., submitted to <i>Restoration Ecology</i>	Sachuest Point — 1997	0.70	1.6	0.5474	1	18.18	0.03
	Roman et al., submitted to <i>Restoration Ecology</i>	Sachuest Point — 1998	0.17	1.6	0.5474	1	18.18	0.01
	Roman et al., submitted to <i>Restoration Ecology</i>	Sachuest Point — 1999	0.55	1.6	0.5474	1	18.18	0.03
	Roman et al., submitted to <i>Restoration Ecology</i>	Sachuest Point — 1998 (lift net)	0.01	1.3	0.5474	1	1.00	0.01
	Roman et al., submitted to <i>Restoration Ecology</i>	Sachuest Point — 1999 (lift net)	0.01	1.3	0.5474	1	1.00	0.01
	Raposa pers comm 2001	NERR — Prudence Isl. Coggeshall — July 2000	2.40	1.6	0.5474	1	19.23	0.11

Table 4-24. Final estimates of the annual increase in production of age-1 equivalent fish per square meter of restored tidal wetland for Pilgrim species that would benefit most from tidal wetland restoration (cont.).

Species	Source of initial species density estimate	Sampling location and date ^a	Reported/ calculated species density estimate per m ² of tidal wetland	Sampling efficiency adjustment factor	Life stage adjustment factor	Restored habitat service flow adjustment factor	Sampling time and location adjustment factor	Increased production of age 1 fish per m ² of restored tidal wetland ^b
Striped killifish	Raposa pers comm 2001	NERR — Prudence Isl. Coggeshall — Sept. 2000	0.53	1.6	0.5474	1	19.23	0.02
	Raposa, in press	Galilee Marsh — 1997	4.35	1.6	0.5474	1	11.90	0.32
	Raposa, in press	Galilee Marsh — 1998	3.50	1.6	0.5474	1	11.90	0.26
	Raposa, in press	Galilee Marsh — 1999	12.40	1.6	0.5474	1	11.90	0.91
	Species average							0.17
Striped bass	no obs.							
Bluefish	no obs							

a. Sampling results are based on collections using 1 m² throw traps unless otherwise noted.

b. Calculated by multiplying the initial species density estimate by the sampling efficiency, life stage, and restored habitat service flow adjustment factors and dividing by the sampling time and location adjustment factor.

4.5.3.2 Adjusting artificial reef sampling results to estimate annual average increase in production of age-1 fish

As with the other restoration alternatives, sampling efficiency, life stage conversion, and restored versus undisturbed habitat adjustments were made to production estimates for artificial reef habitats. These adjustments are discussed below.

Sampling efficiency

The same sampling efficiency adjustment factor of 2.0 is incorporated for the tautog abundance estimates developed from the Rhode Island juvenile finfish survey as was used in the sampling efficiency adjustments from this survey for winter flounder. The 2.0 adjustment factor represents the bottom range (conservative assumption) of a seine net's sampling efficiency (50%), based on the judgment of the current staff of Rhode Island's coastal pond fish survey (personal communication, J. Temple, Rhode Island Division of Fish and Wildlife, 2002).

The sampling efficiency of the baited traps and tagging procedure used in Lawton et al. (2000) was assumed to be 1.0, as the results of the study already incorporate sampling efficiency as reported.

Conversion to the age-1 equivalent life stage

The information used to develop life stage adjustment factors for juvenile fish to age-1 equivalents is presented in Table 4-28 for the Pilgrim species that would benefit most from artificial reef development.

Table 4-28. Life stage adjustment factors for Pilgrim species — artificial reef.

Species	Oldest life stage before age 1 in I&E model	Estimated survival rate to age 1	Sampled life stage	Estimated life stage adjustment factor
Rock gunnel	larvae	0.1416	juvenile	0.5708
Radiated shanny	larvae	0.0853	juvenile	0.5426
Sculpin spp.	larvae	0.0180	juvenile	0.5090
Tautog	larvae	0.0001	juvenile	0.5001

The Rhode Island juvenile finfish survey primarily captures juvenile tautog. However, the size distribution of cunner suggests that primarily adult fish were captured. Some of these cunner were likely older than age 1. To convert the raw cunner numbers to age-1 equivalents, we used the same factor of 1.39 that is also used in the EAM to convert the raw numbers of cunner impinged to age-1 equivalents.

Adjusting for differences between restored and undisturbed habitats

No available information suggested that artificial reefs are utilized substantially less than natural reefs by the species listed in Table 4-25. Thus, an adjustment factor of 1.0 was incorporated.

4.5.3.3 Final estimates of increases in age-1 production for artificial reefs

Table 4-29 presents the final estimates of annual increased production of age-1 fish, based on the average across all sampling efforts, that would result from artificial reef development for species at Pilgrim.

Table 4-29. Final estimates of annual increased production of age-1 equivalent fish per square meter of artificial reef developed for Pilgrim species.

Species	Source of initial species density estimate	Species abundance estimates (fish/m ² reef)	Sampling efficiency adjustment factor	Life stage adjustment factor	Restored vs. undisturbed habitat adjustment factor	Expected age-1 increased production (fish per m ² artificial reef)
Rock gunnel	no obs.					
Radiated shanny	no obs					
Cunner	Lawton et al. (2000), Plymouth MA	4.06 ^a	1.0	1.39	1.0	5.64
Sculpin spp.	No obs.					
Tautog	RI juvenile finfish survey, 1990-2000: Patience Island	0.028	2.0	0.5001	1.0	0.03
	RI juvenile finfish survey, 1990-2000: Spar Island	0.031	2.0	0.5001	1.0	0.03
	Species average					0.03

a. Average of the central population estimates for the inner and outer breakwaters.

4.5.4 Estimates of Increased Species Production from Installed Fish Passageways

A habitat-based option for increasing the production of anadromous species is to increase their access to suitable spawning and nursery habitat by installing fish passageways at currently impassible barriers (e.g., dams). The anadromous species at Pilgrim that would benefit most from fish passageways are presented in Table 4-30, along with information on their annual average I&E losses for the period 1974-1999.

Table 4-30. Anadromous species at Pilgrim that would benefit most from fish passageways.

Species	Annual average I&E loss of age-1 equivalents	Percentage of annual average I&E loss across all fish species
Rainbow smelt	1,330,022	9.23%
Atlantic herring	29,079	0.20%
Alewife	4,343	0.03%
Blueback herring	703	0.00%
White perch	73	0.00%
Total	1,364,220	9.46%

4.5.4.1 Abundance estimates for anadromous species

No studies provided direct estimates of increased production of anadromous fish attributable to the installation of a fish passageway. Thus, increased production estimates were based on abundance estimates from anadromous species monitoring programs in Massachusetts and Rhode Island, combined with an estimate of the average increase in suitable spawning habitat that would be provided upstream of the current impassible obstacles following the installation of fish passageways.

Anadromous species abundance in Massachusetts and Rhode Island spawning/nursery habitats

Information on the abundance of anadromous species in spawning/nursery habitat in Massachusetts was available only for a select number of alewife spawning runs in the area around the Cape Cod canal, including locations in Massachusetts Bay and Buzzards Bay (personal communication, K. Reback, Massachusetts Division of Marine Fisheries, 2001). Alewife abundance information was also available for the spawning runs at the Gilbert Stuart and Nonquit locations in Rhode Island. These runs are almost exclusively alewives, despite being reported as runs of river herring (i.e., blueback herring and alewives; personal communication, P. Edwards, Rhode Island Department of Environmental Management, 2001). The size of these alewife runs and the associated abundance estimates (number of fish per acre) in available spawning/nursery habitat are presented in Table 4-31.

The Mattapoissett system has low spawning habitat utilization by alewives because of continuing recovery of the system (personal communication, K. Reback, Massachusetts Division of Marine Fisheries, 2001). Therefore, the Mattapoissett River values were omitted. This raised the production estimates for fish passageways and reduced the restoration costs for implementing sufficient fish passageways.

Table 4-31. Average run size and density of alewives in spawning nursery habitats in select Massachusetts waterbodies.

Waterbody	Average alewife run size (number of fish)	Average number of fish per acre of spawning/nursery habitat
Back River (MA) (12 year average)	373,608	766
Mattapoissett River ^a (12 year average)	66,457	90
Monument River (MA) (12 year average)	367,521	811
Nonquit system (RI) (1999-2001 average)	192,173	951
Gilbert Stuart system (RI) (1999-2001 average)	311,839	4,586
Average across all sites presented		1,441
Average without Mattapoissett River		1,778

a. The Mattapoissett River is currently in recovery and production has been increasing in recent years (personal communication, K. Reback, Massachusetts Division of Marine Fisheries, 2001).

Average size of spawning/nursery habitat that would be accessed with the installation of fish passageways

Anadromous fisheries staff in Massachusetts revealed that approximately 5 acres of additional spawning/nursery habitat would become accessible for each average passageway installed (personal communication, K. Reback, Massachusetts Division of Marine Fisheries, 2001). This estimate reflects that previous projects have already provided access to most of the available large spawning/nursery habitats.

4.5.4.2 Adjusting anadromous run sampling results to estimate annual average increase in production of age-1 fish

As with the other restoration alternatives, a number of adjustment factors were considered. However, information was much more limited upon which to base these adjustments. Adjustments to convert returning alewives to age-1 equivalents and to account for sampling efficiency were assumed to be 1.0 because of a lack of information. In addition, nothing suggested a basis for adjustments based on differences between existing and new spawning habitat accessed via fish passageways. As a result, an adjustment factor of 1.0 was used.

4.5.4.3 Final estimates of annual age-1 equivalent increased species production

The density of anadromous species in their spawning/nursery habitat, the average increase in spawning/nursery habitat from installation of fish passageways, and adjustment factors are presented in Table 4-32.

Table 4-32. Estimates of increased age-1 fish for Pilgrim species that would benefit most from installation of fish passageways.

Species	Source of initial species density estimate	Species density estimate in spawning/nursery habitat (fish per acre)	Number of additional spawning/nursery habitat acres per new passageway	Life stage adjustment factor	New vs. existing habitat adjustment factor	Calculated annual increase in age-1 fish per new passageway installed ^a
Rainbow smelt	no obs					
Atlantic herring	no obs					
Alewife	Mattapoissett River — (K. Reback MA DMF pers. comm, 2001)	90	5	1	1	452
	Monument River — (K. Reback MA DMF pers. comm, 2001)	810	5	1	1	4,054
	Back River — (K. Reback MA DMF pers. comm, 2001)	766	5	1	1	3,828
	Nonquit river system — (P. Edwards, RI DEM, pers comm, 2001)	951	5	1	1	4,757
	Gilbert Stuart river system — (P. Edwards, RI DEM, pers comm, 2001)	4,586	5	1	1	22,929
	Species average (excluding Mattapoissett River)^b					8,892
Blueback herring	no obs.					
White perch	no obs.					

a. This value is the product of the values in the five data fields.

b. As previously noted, the Mattapoissett results are excluded in calculating the species average for alewife because the low density estimates are attributable to the system recovering from previous stressors.

4.5.5 Estimates of Increase in Age-1 Fish Production from Water Quality Improvements or Reduced Fishing Pressure

Resource managers and restoration experts indicated that a number of Pilgrim species would benefit most from improved water quality or reduced fishing pressure because they met at least one of the following criteria:

- ▶ The species is pelagic (e.g., Atlantic menhaden).
- ▶ There is no obvious habitat that the species prefers or relies on that could be practically restored (e.g., hogchoker).
- ▶ The preferred habitat is in deep water (e.g., greater than 30 feet) or very deep water (e.g., greater than 100 feet), which limits practical options for habitat restoration because of cost or technical constraints (e.g., fourbeard rockling, American plaice).

As a result, pursuing improvements in water quality and/or reducing fishing pressure were selected as the preferred restoration alternatives for these species. The species at Pilgrim that would benefit most from improving water quality or reducing fishing pressure are listed in Table 4-33, along with annual average I&E losses for the period 1974-1999.

Table 4-33. Pilgrim species that would benefit most from improving water quality or reducing fishing pressure.

Species	Average annual I&E loss of age-1 equivalent organisms	Percentage of total I&E losses for all species
Finfish		
Fourbeard rockling	411,191	2.85%
Windowpane	17,542	0.12%
Atlantic menhaden	14,270	0.10%
Atlantic mackerel	6,662	0.05%
Searobin	3,767	0.03%
Red hake	1,774	0.01%
Lumpfish	1,297	0.01%
Butterfish	399	0.00%
American plaice	221	0.00%
Scup	114	0.00%
Little skate	78	0.00%
Bay anchovy	18	0.00%
Hogchoker	2	0.00%
Total	457,335	3.17%
Shellfish		
Blue mussels	159,880,528,203	100%

Despite the magnitude of I&E losses for these species, and the fact that improving water quality and reducing fishing pressure would benefit all species to varying degrees, it was beyond the scope of this HRC to develop quantitative estimates of the increased production of age-1 fish from these two alternatives. This reflects both budget constraints and a lack of readily available information describing how much water quality projects would improve water quality, and how much water quality improvements would increase fish production. In addition, significant uncertainty exists regarding the effectiveness of nonregulatory actions that could be undertaken to reduce fishing pressure. The limits to developing quantitative estimates of the increased production of age-1 fish are reviewed in the following subsections.

4.5.5.1 Limits to quantifying age-1 production increases from water quality improvements

Several actions could improve water quality without transferring legal responsibility from one party to another. For example, buffer strip development along waterways and septic system improvements would reduce loadings of suspended solids and nutrients into water bodies, improving turbidity, dissolved oxygen content, and chemical concentrations. These improvements could be linked to increases in age-1 fish directly by reducing mortality, or indirectly by stimulating increased natural production.

The expected average annual increases in fish production associated with these restoration actions were not quantified because developing or interpreting complex water quality, concentration-response, and population models was beyond the scope of this HRC valuation. However, these relationships could be developed with additional time and effort.

4.5.5.2 Limits to quantifying increased species production from reduced fishing pressure

Most actions that can achieve lasting reductions in fishing pressure require changes in existing regulations. However, regulatory changes were beyond the scope of this HRC valuation, particularly because of the uncertainty concerning the lack of established property rights for individual fish. Absent these rights, which could be established through individual allocations of a fixed quota on commercial and recreational catches, reducing fishing pressure on a species generally involves persuading current participants in the fishery to cease or reduce their operations.

While market-based programs such as commercial boat buy-backs (Kitts and Thunberg, 1998) have been implemented to reduce fishing pressure, their impact is uncertain because these boats generally have an operating license that permits a limited number of days at sea or other level of effort. While this limits the number of days at sea for a given fleet, its impact may be minimal if the most productive boats remain in the fleet. Further, removing the effort of a given boat may have little impact if it was not actively fishing or if the remaining vessels increase their level of effort. For these reasons the potential benefits of reduced fishing pressure were not quantified.

4.6 Step 6: Scaling Preferred Restoration Alternatives

The following subsections calculate the required scale of implementation for each of the preferred restoration alternatives for each species. The quantified I&E losses are divided by the estimates of the increased fish production, giving the total amount of each restoration needed to offset I&E losses for each species.

4.6.1 SAV Scaling

The information used to scale SAV restoration is presented in Table 4-34.

Table 4-34. Scaling of SAV restoration for Pilgrim species.

Species	Average annual I&E loss of age-1 equivalent fish	Best estimate of increased production of age-1 fish per 100 m ² of revegetated substrate (rounded)	Number of 100 m ² units of revegetated SAV required to offset estimated average annual I&E loss
Atlantic tom cod	2,439	0.99	2,475
Pollock	525	no obs.	N/A
Northern pipefish	118	2.50	47
Threespine stickleback	118	19.29	6
Required units of implementation to offset I&E losses across species			2,475

4.6.2 Tidal Wetlands Scaling

The information used to scale tidal wetland restoration is presented in Table 4-35.

Table 4-35. Scaling of tidal wetland restoration for Pilgrim species.

Species	Average annual I&E loss of age-1 equivalent fish	Best estimate of increased production of age-1 fish per m ² of restored tidal wetland (rounded)	Number of m ² units of restored tidal wetland required to offset estimated average annual I&E loss ^a
American sand lance	4,116,285	no obs.	N/A
Winter flounder	210,715	0.09	2,429,812
Atlantic silverside	25,929	0.19	139,539
Grubby	879	no obs.	N/A
Striped killifish	90	0.17	527
Striped bass	9	no obs.	N/A
Bluefish	2	no obs.	N/A
Required units of implementation to offset I&E losses across species			2,429,812

a. A restored wetland area refers to an area in a currently restricted tidal wetland where invasive species (e.g., *Phragmites* spp.) have overtaken salt tolerant tidal marsh vegetation (e.g., *Spartina* spp.) and that is expected to revert to typical tidal marsh vegetation once tidal flows are returned. Waterways adjacent to these vegetated areas are also included in calculating the potential area that could be restored in a tidal wetland.

4.6.3 Reef Scaling

The information used to scale artificial reef development is presented in Table 4-36.

Table 4-36. Scaling of artificial reef development for Pilgrim species.

Species	Average annual I&E loss of age-1 equivalent fish	Best estimate of increased production of age-1 fish per m ² of artificial reef (rounded)	Number of m ² units of artificial reef surface habitat required to offset estimated average annual I&E loss
Rock gunnel	4,862,872	no obs.	N/A
Radiated shanny	1,644,456	no obs.	N/A
Cunner	993,911	5.64	176,218
Sculpin species	734,773	no obs.	N/A
Tautog	1,076	0.03	36,699
Required units of implementation to offset I&E losses across species			176,218

4.6.4 Anadromous Fish Passage Scaling

The information used to scale fish passageway installation is presented in Table 4-37.

Table 4-37. Scaling of anadromous fish passageways for Pilgrim species.

Species	Average annual I&E loss of age-1 equivalent fish	Best estimate of increased production of age-1 fish per passageway installed (rounded)	Number of new fish passageways required to offset estimated average annual I&E loss
Rainbow smelt	1,320,022	no obs.	N/A
Atlantic herring	29,079	no obs.	N/A
Alewife	4,343	8,892	0.49
Blueback herring	703	no obs.	N/A
White perch	73	no obs.	N/A
Required units of implementation to offset I&E losses across species			0.49

4.6.5 Water Quality Improvement/Reduce Fishing Pressure Scaling

It was not possible to scale sufficient water quality improvements and reduced fishing pressure to offset I&E losses. The Pilgrim species that would benefit most from improving water quality and reducing fishing pressure are presented in Table 4-38. Scaling this restoration alternative likely would increase the Pilgrim HRC estimate significantly, as discussed in Section 4.9.

Table 4-38. Pilgrim species that would benefit most from improved water quality/reduced fishing pressure.

Species	Average annual I&E loss of age-1 equivalent fish	Best estimate of increased production of age-1 fish from water quality/reduced fishing pressure improvements	Number of units of water quality improvement required to offset estimated average annual I&E loss
Finfish			
Fourbeard rockling	411,191	no obs.	N/A
Windowpane	17,542	no obs.	N/A
Atlantic menhaden	14,270	no obs.	N/A
Atlantic mackerel	6,662	no obs.	N/A
Searobin	3,767	no obs.	N/A
Red hake	1,774	no obs.	N/A
Lumpfish	1,297	no obs.	N/A
Butterfish	399	no obs.	N/A
American plaice	221	no obs.	N/A
Scup	114	no obs.	N/A
Little skate	78	no obs.	N/A
Bay anchovy	18	no obs.	N/A
Hogchoker	2	no obs.	N/A
Shellfish			
Blue mussel	159,880,528,203	no obs.	N/A

4.7 Unit Costs

The seventh step of the HRC valuation is to develop unit cost estimates for the restoration alternatives. Unit costs account for all the anticipated expenses associated with the actions required to implement and maintain restoration. Unit costs also included the cost of monitoring to determine increased production of age-1 fish. Unit costs were expressed as the current level of funding required to cover all expenses over the anticipated project life.

All major project expenditures were assumed to occur in the first year, leaving only maintenance and monitoring expenses in subsequent years. Most of these projects were assumed to require little or no maintenance. The monitoring programs were assumed to last for 10 years. Therefore, the current funding required for a unit of each restoration alternative was calculated as the sum provided at the project outset that could fund all activities for 10 years, accounting for inflation and interest. The following price inflation and interest earnings assumptions were made:

- ▶ An annual price inflation rate of 3.0% was used, consistent with the observed annual rate in the Consumer Price Index from 1990 through 2000 (U.S. Bureau of Labor Statistics, 2001).
- ▶ Interest earnings were calculated by multiplying remaining balances at the end of each year by the estimated December 2001 Treasury bill rate of 5.16% (U.S. Bureau of Housing and Urban Development, 2001).

4.7.1 Unit Costs of SAV Restoration

Unit cost estimates for SAV restoration were expressed as the present value of costs per 100 m² for direct comparison with increased production estimates. A number of completed and ongoing SAV restoration projects were evaluated, and monitoring costs were included. The following subsections describe how implementation and monitoring costs were derived for SAV restoration.

4.7.1.1 Implementation costs

Save the Bay has a long history of SAV habitat assessment and restoration in the Naragansett and Mount Hope Bays. A Save the Bay SAV restoration project begun in the summer of 2001 involved transplanting eelgrass to revegetate 16 m² of habitat at each of three sites in Naragansett Bay. Cost information from this project was used to develop unit cost estimates for implementing SAV restoration per 100 m² of revegetated habitat.

Save the Bay's cost proposal estimated that \$93,128 (2001 dollars) would be required to collect and transplant eelgrass shoots over 48 m² of revegetated habitat. These costs include collecting and transplanting the SAV shoots to provide an initial density of 400 shoots per revegetated square meter of substrate. Averaged over the 48 m² of habitat being revegetated, this provides an average unit cost of \$1,940 per m². The unit costs comprise the following categories:

- ▶ labor: 70.7% (includes salaried staff with benefits, consultants, and accepted rates for volunteers)
- ▶ boats: 15.2% (expenses for operating the boat for the collecting and transplanting)
- ▶ materials and equipment: 9.6%
- ▶ overhead: 4.6% (calculated as a flat percentage of the labor expenses for the salaried staff).

Contingency expenses were set at 10% (\$194 per m²). The costs of identifying and evaluating the suitability of potential restoration sites were set at 1% (\$19 per m²). No costs were added for maintaining the service flows provided by the project, because SAV restoration requires little direct maintenance. This reflects both the relative inaccessibility of SAV sites and the relative importance of factors beyond direct control, such as local water quality and extreme weather.

Costs were also adjusted to account for natural growth and spreading from the original transplant sites to the bare spots between transplants (Short et al., 1997). For example, Dr. Frederick Short (University of New Hampshire's Jackson Estuarine Laboratory) planted between 120 and 130 TERFS (Transplanting Eelgrass Remotely with Frame Systems), each 1 m², in each acre of seabed to be revegetated at a SAV restoration site (personal communication, P. Colarusso, U.S. EPA Region 1, 2002). Assuming complete coverage over time, this results in a ratio of plantings to total coverage of between 1:31 (130 1 m² TERFS / 4,047 m² per acre) and 1:34 (120 1 m² TERFS / 4,047 m² per acre).

However, the initially bare areas do not revegetate immediately. Therefore, an assumption was made that the area covered would double each year. Under this assumption, the entire area would be filled in the sixth year of the restoration project. Using the habitat equivalency analysis (HEA) method (Peacock, 1999), the present value of the services over the 6 years is 90% of that provided by instantaneous revegetation. Therefore, 90% of the 1:34 planting-to-coverage ratio, or 1:30, was applied. Table 4-39 presents the components of implementation unit cost for SAV restoration, incorporating the adjustment ratio in the last step.

Table 4-39. Implementation unit costs for SAV restoration.

Expense category	Cost per m ² of SAV restored	Cost per 100 m ² of SAV restored
Direct restoration (shoot collection and transplant)	\$1,940	\$194,000
Contingency costs (10% of direct restoration)	\$194	\$19,400
Restoration site assessment (1% of direct restoration)	\$19	\$1,900
Subtotal without allowance for distribution of transplanted SAV shoots	\$2,154	\$215,400
Discounted rate of return on transplanted SAV	30:1	30:1
Final implementation unit costs	\$71.80	\$7,180

4.7.1.2 Monitoring costs

SAV restoration monitoring improves the inputs to the HRC analysis by quantifying the impact of the SAV restoration on fish production/recruitment in the restoration area, and the rate of growth and expansion of the restored SAV bed. The most efficient way to achieve both of these goals would be for divers to evaluate the number of adult fish in the habitat and the vegetation density, combined with throw trap or drop trap sampling of juvenile fish using the habitat (Short et al., 1997). Diver-based monitoring minimizes damage to sites, expands the areas that can be sampled, and increases sampling efficiency compared to trawl-based monitoring (personal communication, J. Hughes, NOAA Marine Biological Laboratory, 2001).

Hourly rates for the divers and captain were provided by Save the Bay (personal communication, A. Lipsky, Save the Bay, 2001), and the daily rate for the boat was based on rate information from NOAA's Marine Biological Laboratory in Woods Hole (personal communication, J. Hughes, NOAA, 2001). Because SAV monitoring costs will be significantly affected by the size, number, and distance between restored SAV habitats, large areas can be covered in a single day only when continuous habitats are surveyed. Smaller, disconnected habitats will require much more time to cover. Therefore, total monitoring costs are somewhat unpredictable and were assumed to be equal to initial revegetation costs. This simplifying assumption is neither conservative, nor liberal. The summary of the available SAV monitoring costs and the final assumption used are presented in Table 4-40.

Table 4-40. Estimated annual unit costs for a SAV restoration monitoring program.

Annual expenditures			
Expense category	Quantity	Daily rate	Total cost
Monitoring crew	3 (2 divers and boat captain/assistant)	\$268	\$804
Monitoring boat	1	\$150	\$150
Total daily rate			\$954
Assumed PV cost for SAV monitoring per 100 m ² restored habitat			\$7,180

4.7.1.3 Total SAV restoration costs

Combining the unit costs for restoration and monitoring, the cost for a 100 m² unit of SAV restoration for 10 years is \$14,360.

4.7.2 Unit Costs of Tidal Wetland Restoration

Many different actions may be needed to restore flows to a wetland site, and project costs can vary widely. These issues are addressed in the following subsections, which present the development of the unit costs for tidal wetland restoration.

4.7.2.1 Implementation costs

Costs for restoration of tidally restricted marshes depend heavily on the type of restriction that is impeding tidal flow into the wetland. Possible sources of the restriction in tidal flow include improperly designed or located roads, railroads, bridges, and dikes, all of which can eliminate tidal flows or restrict tidal flows via improperly sized openings. A compilation of tidally restricted salt marsh restoration projects in the Buzzards Bay watershed (Buzzards Bay Project National Estuary Program, 2001) describes restrictions and costs to return tidal flows to over 130 sites. These cost estimates include expenses for project design, permitting, and construction, and are estimated on a predictive cost equation that was fitted from the actual costs and budgets for a limited number of projects (Buzzards Bay Project National Estuary Program, 2001).

Staff involved in the Buzzards Bay assessment provided the current project database, which includes the following information (personal communication, J. Costa, Buzzards Bay National Estuary Program, 2001):

- ▶ nature of the tidal restriction
- ▶ estimated cost to address the tidal restriction
- ▶ size of the affected tidal wetland (in acres)
- ▶ acreage of the *Phragmites* in the tidally restricted wetland.

Some of the project costs used in the cost estimation equation were provided by public agencies, which were lower than market prices (personal communication, J. Costa, Buzzards Bay National Estuary Program, 2001). Therefore, the cost estimates were adjusted upward by a factor of 2.0, consistent with the adjustment recommended in the report (Buzzards Bay Project National Estuary Program, 2001). The adjusted total project costs were then divided by the acres of *Phragmites* in the wetland to provide the cost per acre (sites with no *Phragmites* were eliminated from consideration). Table 4-41 summarizes costs based on the cost factor (an input in the cost estimation equation), type of restriction found at the site, and the number of *Phragmites* acres at the location. An alternative summary of these projects is presented in Table 4-42, where the projects are organized by acres of *Phragmites* at the site, not the current tidal restriction.

Combined, Tables 4-41 and 4-42 show significant variability in the per acre costs for tidal wetland restoration. Therefore, the median cost of \$71,000 per acre of tidal wetland restoration was used. Table 4-43 presents the final per acre implementation costs for tidal wetland restoration. These costs include the median per acre restoration cost, \$750 per acre, paid by the Rhode Island Department of Environmental Management's Land Acquisition Group for this type of land (personal communication, L. Primiano, Rhode Island Department of Environmental Management, 2001).

Table 4-41. Salt marsh restoration costs.

Restriction structure class	Cost factor	Phragmites acres	Number of sites	Cumulative Phragmites acreage	Average Phragmites acreage	Total private cost	Average cost per Phragmites acre restored (from total cost and acres)	Minimum cost per Phragmites acre restored	Maximum cost per Phragmites acre restored
culvert	0.5	acres < 1	16	6.59	0.41	\$335,357	\$50,889	\$17,921	\$578,081
culvert	0.5	1 < acres < 5	11	20.37	1.85	\$242,496	\$11,903	\$3,242	\$71,045
culvert	0.5	5 < acres < 10	1	8.56	8.56	\$20,825	\$2,434	\$2,434	\$2,434
dike	0.5	acres < 1	1	0.35	0.35	\$13,211	\$38,073	\$38,073	\$38,073
road	0.5	1 < acres < 5	1	1.67	1.67	\$19,116	\$11,447	\$11,447	\$11,447
culvert	1	acres < 1	31	13.26	0.43	\$1,797,450	\$135,585	\$21,518	\$10,490,647
culvert	1	1 < acres < 5	23	46.02	2.00	\$1,225,745	\$26,633	\$5,312	\$84,770
culvert	1	5 < acres < 10	2	16.43	8.22	\$248,878	\$15,144	\$9,898	\$22,608
culvert	1	10 < acres < 25	2	41.97	20.99	\$91,451	\$2,179	\$1,919	\$2,449
dike	1	10 < acres < 25	1	12.00	12.00	\$6,053,000	\$504,417	\$504,417	\$504,417
fill	1	acres < 1	1	0.12	0.12	\$31,142	\$251,146	\$251,146	\$251,146
road	1	acres < 1	1	0.10	0.10	\$29,396	\$293,958	\$293,958	\$293,958
road	1	1 < acres < 5	1	2.31	2.31	\$35,231	\$15,265	\$15,265	\$15,265
wall	1	acres < 1	2	0.96	0.48	\$148,819	\$154,697	\$25,661	\$5,936,752
bridge	3	acres < 1	8	5.12	0.64	\$21,208,029	\$4,140,576	\$184,170	\$13,418,293
bridge	3	1 < acres < 5	12	27.32	2.28	\$27,704,691	\$1,014,192	\$184,048	\$3,663,062
bridge	3	5 < acres < 10	2	11.01	5.51	\$6,606,000	\$599,946	\$399,746	\$800,545
bridge	3	10 < acres < 25	8	103.49	12.94	\$92,094,000	\$889,883	\$56,300	\$3,300,250
bridge	3	25 < acres < 50	4	157.28	39.32	\$8,262,000	\$52,529	\$22,882	\$105,968
bridge	3	50 < acres	1	113.00	113.00	\$6,163,000	\$54,540	\$54,540	\$54,540
railroad	4	acres < 1	1	0.41	0.41	\$66,841	\$163,826	\$163,826	\$163,826
railroad	4	1 < acres < 5	3	3.61	1.20	\$1,078,692	\$298,476	\$208,033	\$13,418,293

Table 4-42. Average per acre cost of restoring *Phragmites* in Buzzards Bay restricted tidal wetlands.

<i>Phragmites</i> acres	Number of sites	Cumulative acreage	Average acreage	Total private cost	Average cost per <i>Phragmites</i> acre restored (from total cost and acres)
acres < 1	61	26.91	0.44	\$23,630,245	\$878,121
1 < acres < 5	51	101.31	1.99	\$30,305,971	\$299,153
5 < acres < 10	5	36.00	7.20	\$6,875,703	\$190,992
10 < acres < 25	11	157.46	14.31	\$98,238,451	\$623,895
25 < acres < 50	4	157.28	39.32	\$8,262,000	\$52,529
50 < acres	1	113.00	113.00	\$6,163,000	\$54,540
Total	133	591.96	4.45	\$173,475,370	\$293,053
Median					\$71,000

Table 4-43. Implementation unit costs for tidal wetland restoration incorporated in the HRC.

Implementation cost description	Source of estimate	Value (2001 dollars)
Restore tidal flows to restricted areas	Median of adjusted costs from Buzzards Bay project database	\$71,000
Acquire tidal wetlands	Midpoint of range of paid for tidal wetlands by Rhode Island DEM	\$750

4.7.2.2 Monitoring costs

Neckles and Dionne (1999) present a sampling protocol, developed by a workgroup of experts, for evaluating nekton use in restored tidal wetlands. The sampling plan calls for different sampling techniques and frequencies to capture fish of various sizes in both creek and flooded marsh habitats of a tidal wetland. A summary of these recommendations is presented in Table 4-44.

Table 4-44. Sampling guidelines for nekton in restored tidal wetlands.

Sampling location	Sampling technique	Sampling time	Sampling frequency
Creeks (for small fish)	Throw traps	midtide during spring tide cycle	2 dates in August
Creeks (for larger fish)	Fyke net	slack tide during spring tide cycle	2 dates in August (same as for throw trap work) and 2 dates in spring
Flooded wetland surface	Fyke net	spring tide cycle	1 date in August

Source: Neckles and Dionne, 1999.

The sampling protocol suggests that one technician and two volunteers can provide the necessary labor. The estimated annual cost in the first year of monitoring is \$1,600. This cost comprises \$490 in labor for the three workers over 5 days (3 in August and 2 in the spring, with 8-hour days, \$15 per hour for volunteers, and \$30 per hour for the technician). The \$1,100 in equipment costs includes two fyke nets and two throw traps at \$500 for the fyke nets and \$50 for homemade throw traps (Neckles and Dionne, 1999). Two sets of this sampling equipment would allow simultaneous sampling in a restored marsh and at a reference location. Treating these costs as a per acre cost for aggregation with implementation costs probably overstates the frequency of sampling required at the site. However, the initial year labor cost of \$500 per acre has little impact compared to implementation and overall costs.

4.7.2.3 Total tidal wetland restoration costs

Combining implementation and monitoring costs for tidal wetland restoration with annual price inflation (3%) and interest earned on balances carried over (5.16%), the cost for an acre of tidal wetland restoration is \$78,500, or \$19 per m², which was used in the development of the total Pilgrim HRC valuation.

4.7.3 Artificial Reef Unit Costs

The unit cost estimates for developing and monitoring artificial reefs are based the construction and monitoring of six 30 ft x 60 ft reefs constructed of 5-30 cm diameter stone in Dutch Harbor, Naragansett Bay (personal communication, J. Catena, NOAA Restoration Center, 2001). While these reefs were constructed for lobsters, surveys of the Dutch Harbor reef have noted abundant fish use of the structures (personal communication, K. Castro, University of Rhode Island, 2001).

4.7.3.1 Implementation costs

The summary cost information for the design and construction of the six reefs in Dutch Harbor is presented in Table 4-45 (personal communication, J. Catena, NOAA Restoration Center, 2001).

Table 4-45. Summary cost information for six artificial reefs in Dutch Harbor, Rhode Island.

Project component	Cost
Project design	not explicitly valued, received as in-kind services
Permitting	not explicitly valued, received as in-kind services
Interagency coordination	not explicitly valued, received as in-kind services
RFP preparation	not explicitly valued, received as in-kind services
Contract management	not explicitly valued, received as in-kind services
Baseline site evaluation	\$12,280
Reef materials (600 yd ³ of 2-12 in. stone)	\$12,000
Reef construction	\$35,400
Total	\$59,680

These costs were converted to cost per square meter of surface habitat. The cumulative surface area of the six reefs, assuming that the reefs have a sloped surface on both sides, and based on the volume of material used, is approximately 1,024 m². Dividing the total project costs by this surface area results in an implementation cost of \$58/m² of artificial reef habitat.

4.7.3.2 Monitoring costs

Monitoring costs for the Dutch Harbor reefs were \$140,000 over a 5 year period. Again, assuming similar assessment techniques would be required to evaluate fish use and production of an artificial reef (i.e., diver surveys and trap work), these costs are adjusted to provide a monitoring expense of \$28,000.

4.7.3.3 Total artificial reef costs

Combining costs for implementation and monitoring of an artificial reef with annual price inflation (3%) and the interest earned on balances carried over (5.16%), the cost is \$308/m² (\$315,167/1,024 m² surface area over the six reefs), which was used in the development of the total Pilgrim HRC valuation.

4.7.4 Costs of Anadromous Fish Passageway Improvements

Unit costs for fish passageways were developed from a series of budgets for prospective anadromous fish passageway installation, combined with information provided by staff involved with anadromous species programs in Massachusetts and Rhode Island. The implementation, maintenance, and monitoring costs for a fish passageway are presented in the following subsections.

4.7.4.1 Implementation costs

Projected costs for four new Denil type fish passageways on the Blackstone River at locations in Pawtucket and Central Falls, Rhode Island, provide the base for the implementation cost estimates for anadromous fish passageways (personal communication, T. Ardito, Rhode Island Department of Environmental Management, 2001). The reported lengths of the passageways in these projects ranged from 32 m to 82 m, with associated changes in vertical elevation ranging from slightly more than 4 m to approximately 10 m based on the reported slope ratios of 1:8.

The average cost for these projects was \$513,750. The average cost per meter of passageway length was \$10,300 and per meter of vertical elevation covered was \$82,600. These estimates are consistent with the approximate values of \$9,800 per meter of passageway length and \$98,000 per vertical meter suggested by the U.S. Fish and Wildlife Service's regional Engineering Field Office (personal communication, D. Quinn, U.S. Fish and Wildlife Service, 2001). An alternative style of fish passageway, the Alaskan steep, has lower unit costs of \$33,000 per vertical meter, but is not suited for many locations. Therefore, its costs were not used to develop implementation unit cost estimates. While all parties contacted noted that fish passageway costs are extremely sensitive to local conditions, this HRC valuation uses the estimate of \$513,750 as its basic implementation unit cost for installing an anadromous fish passage, assuming the characteristics of the four sites on the Blackstone River are representative of the conditions that would be found at other suitable locations for new passageways.

4.7.4.2 Maintenance and monitoring costs

Maintenance requirements for the Denil fish passageway are minimal and generally consist of periodic site visits to remove any obstructions, typically with a rake or pole (personal communication, D. Quinn, U.S. Fish and Wildlife Service, 2001). Denil passageways located in Maine are still functioning after 40 years, so no replacement costs were considered as part of the maintenance for the structure. Monitoring a fish passageway consists of installing a fish counting monitor and retrieving its data.

A new fish passageway would be visited three times a week during periods of migration (personal communication, D. Quinn, U.S. Fish and Wildlife Service, 2001). Each site visit would require 2 hours of cumulative time during 8 weeks of migration. Volunteer labor costs \$15/hr. Therefore, the annual cost for labor in the first year would be \$740. The cost of a fish counter is \$5,512, based on the average price of two fish counters listed by the Smith-Root Company (Smith-Root, 2001).

4.7.4.3 Total fish passageway unit costs

Combining the costs for implementation, maintenance, and monitoring of an anadromous fish passageway with the annual price inflation (3%) and the interest earned on balances carried over (5.16%), the cost of a single new Denil type fish passageway is \$526,000.

4.7.5 Unit Costs for Water Quality Improvements/Reductions in Fishing Pressure

Because increased fish production from water quality improvements or reduced fishing pressure was not calculated, unit costs were not determined for this restoration option. However, examples of water quality improvement projects were summarized to provide a sense of the potential magnitude of costs. The costs of a commercial boat buyback program to reduce fishing pressure on various Northeast groundfish stocks were also summarized. The cost summaries are presented in the following subsections.

4.7.5.1 Cost information from a select set of water quality improvement projects

Table 4-46 provides information from several water quality improvement projects in coastal areas between Massachusetts Bay and Narragansett Bay that address nutrient and bacterial pollution resulting from sanitary waste and other anthropogenic sources. Table 4-46 also shows a wide range of water quality projects involving a wide range of water quality impacts. These projects represent only a few of the projects that could improve water quality in the waters from Massachusetts Bay to Narragansett Bay. Existing project proposals could easily cost billions of dollars.

Table 4-46. Examples of nonpoint source pollution restoration projects in Massachusetts.

Project	Location	Goals	Tasks	Total Cost
Combined sewer overflow (CSO) upgrade ^a	Naragansett Bay, Providence, Pawtucket, and Central Falls, RI	Treatment of ~2.2 billion gallons of waste that are discharged untreated into the bay each year from the combined sewer overflows.	Construct 6 miles of underground storage tunnels, two sedimentation/disinfection treatment facilities, one wetland treatment system, and sewer separation of 12 areas.	\$389,000,000
Septic system improvements ^b	Bluefish River, Duxbury, MA	Opened soft-shelled clam beds over approximately one-half mile of the river to shellfishing.	Connected septic systems from 3 historic homes and 19 commercial properties on the river to a centralized leach field outside of the river basin.	\$800,000
Stormwater treatment ^c	Onset Bay, Wareham, MA	Part of a series of water quality improvement projects aimed at upgrading seasonally closed shellfishing areas and reducing discharges along public beaches.	Design and construct stormwater remediation best management practices (BMPs) for four stormwater outfalls. Develop a quality assurance plan and perform pre- and post-construction water quality monitoring. Conduct public outreach programs and workshops.	\$218,000
Treatment of road runoff ^c	Three Bay Area/Ropes Beach, Barstable, MA	Protection of Cotuit Bay, a shellfishing area, and gateway to two anadromous fish runs, from nutrient and sediment loading.	Design and install sediment removal tanks, an infiltration system, and a series of rock filled pools and channels to remove sediment bacteria and nitrogen from road runoff contributing to contamination of Cotuit Bay. Develop a quality assurance plan and conduct monitoring. Conduct a technology transfer presentation.	\$157,050
Stormwater treatment ^c	First Herring Brook, Scituate, MA	Protect a pond that supplies the town's water supply from contamination.	Disconnect 9 stormwater discharges in a highly developed area and install infiltration BMPs. Develop a quality assurance plan and conduct monitoring. Make system design to other local developers.	\$129,300
Parking lot runoff treatment ^c	Shaw's Plaza, Sharon, MA	Improve water quality in Billing's Brook and in nearby wetlands and public water supply wells.	Develop and implement stormwater BMPs, including a drainage system with an oil/gas separator catch basin and infiltrations. Develop a maintenance program to ensure that it functions properly. Initiate a public education program on the potential impacts of pollution from runoff from roads and parking lots.	\$48,000

a. NBC, 2001.

b. Personal communication, Joe Grady, Town of Duxbury, 12/07/01.

c. MADEP, 2000.

4.7.5.2 Cost information for commercial boat buyback program

A demonstration of a commercial boat buyback program was conducted in the Northeast groundfish fishery. Permit-holding boat owners were asked to submit a price at which they would be willing to retire their vessel from fishing and relinquish all their existing fishing permits (Kitts and Thunberg, 1998). These bids were then ranked in ascending order based on the ratio of their bid to the groundfish revenue from reported landings by the boat to maximize the impact of the program (i.e., remove the productive boats first).

From June 1995 through May 1998, 79 boats were bought out and retired from commercial fishing at an average price of roughly \$309,000, with a range from \$50,000 to \$1.1 million (Kitts and Thunberg, 1998). On average, permits that allocated 152.9 days at sea per boat, although the average boat was only using 111.8 of these days (Kitts and Thunberg, 1998). The impact of this program on increased production was not quantified.

4.8 Total Cost Estimation

The eighth and final step in the HRC valuation is to estimate the total cost for the preferred restoration alternatives by multiplying the required scale of implementation for each restoration alternative by the complete unit cost for that alternative. The cost of each restoration alternative was sufficient to offset the I&E losses of all Pilgrim species that benefit most from that alternative (i.e., each restoration type was sufficient to offset the single species with the greatest restoration need for that preferred restoration; however, the restoration needs of all species preferring that habitat were not summed because the same habitat benefits each of the species simultaneously). The costs of each restoration program were then summed to determine the total HRC necessary to offset all Pilgrim losses (i.e., multiple restoration programs were required to benefit the diverse species lost at Pilgrim).

The total HRC estimates for the Pilgrim facility are provided in Table 4-47, along with the species requiring the greatest level of implementation of each restoration alternative to offset I&E losses. The scale of implementation, unit costs, and total costs in this table have been rounded to two significant digits to avoid false precision. Resulting total costs also carry two significant digits. These costs can be converted to annualized values by specifying a time period and interest rate.

Table 4-47. Total HRC estimates for Pilgrim I&E losses.

Preferred restoration alternative	Species requiring the greatest level of restoration implementation		Required units of restoration implementation	Units of measure for preferred restoration alternative	Unit cost	Total cost
	Species	Average annual I&E loss of age-1 equivalents				
Improve water quality/reduce fishing pressure	Fourbeard rockling	411,191	N/A	N/A	N/A	N/A
	Blue mussel	159 billion	N/A	N/A	N/A	N/A
Install fish passageways	Alewife	4,343	0.49	new fish passageway	\$530,000	\$530,000 ^a
Create artificial reefs	Cunner	993,911	180,000	m ² of reef surface area	\$310	\$56,000,000
Restore SAV	Atlantic cod	2,439	2,500	100 m ² of directly revegetated substrate	\$14,000	\$35,000,000
Restore tidal wetland	Winter flounder	210,715	2,400,000	m ² of restored tidal wetland	\$19	\$46,000,000
Total HRC						\$140,000,000

a. Anadromous fish passageways must be implemented in whole units, and increased production data are lacking for most affected anadromous species. Therefore, one new passageway was assumed to be warranted.

4.9 Conclusions

HRC analyses indicate that the present value of minimizing I&E at the Pilgrim CWIS is at least \$140 million. This value is significantly greater than the \$6-7 million (7% interest rate, in perpetuity) of foregone recreational and commercial fishing calculated in the Pilgrim case study for EPA's Section 316(b) rule. Recreational and commercial fishing values are lower primarily because they include only a small subset of species, life stages, and human use services that can be linked to fishing. In contrast, the HRC valuation is capable of valuing all species and life stages, and inherently addresses all of the ecological and public services derived from organisms included in the analyses, even when the services are difficult to measure or poorly understood. However, data gaps, time constraints, and budgetary constraints prevented this HRC valuation from addressing most of the aquatic organisms lost to I&E at the Pilgrim facility. In particular, annual losses of 160 billion blue mussels and 460,000 fish comprising 13 species were not included in this HRC valuation, even though water quality improvements are feasible, cost-effective, and most likely able to offset some or all of the I&E losses of these species at Pilgrim. In addition, data gaps for species that were included in the HRC valuation forced many conservative assumptions that most likely underestimated the cost of fully offsetting many I&E losses.

In addition to broadening the species, life stages, and services valued, the Pilgrim HRC valuation provides a roadmap for mitigating I&E losses residual to permitted technologies, and for improving the HRC analyses by closing critical data gaps through effective monitoring. Many of the species experiencing I&E losses at Pilgrim can benefit from tidal wetland, SAV, reef, and fish passage restorations. Careful monitoring of increased production of target species at restoration sites would improve the Pilgrim HRC valuation, and would make HRC valuations at other sites more reliable. Further, HRC restoration monitoring needs align public, Agency, and facility motives. Effective restorations with reliable data can broaden the Agency's analyses of public losses. Effective restorations with reliable data can increase the production of fish per restoration dollar spent by a facility. The public benefits both from additional BTA options justified by more comprehensive valuation and from effective restorations in the natural environment.

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